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Effects of Recreational Activity and Livestock Grazing on Habitat Use by Breeding Birds in Cottonwood Forests Along the South Fork Snake River

by
Victoria A. Saab

EFFECTS OF RECREATIONAL ACTIVITY AND LIVESTOCK GRAZING ON HABITAT USE BY BREEDING BIRDS IN COTTONWOOD FORESTS ALONG THE SOUTH FORK SNAKE RIVER

by

Victoria A. Saab
USDA, Forest Service
Rocky Mountain Research Station
316 E. Myrtle St.
Boise, Idaho 83702

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University of Colorado, Boulder

PREFACE

The study reported here was part of a larger research effort to evaluate the influence of surrounding landscapes and local land-use practices on habitat relationships of breeding birds in cottonwood riparian forests (Saab 1996). The intent of these studies was to provide information to managers on habitat features that are necessary for the long-term persistence of small landbirds breeding in cottonwood forests. Data presented in this report are based on the distribution and abundance of breeding birds in relation to local land-use practices of livestock grazing and recreational activities, while a report is forthcoming that examines the effects of these practices on nesting success of small landbirds. The influence of surrounding landscapes on habitat use by breeding birds is reported in Saab (1999). In that paper, the relative importance of several spatial scales to habitat selection by birds is examined, including the landscape scale (composition and patterning of surrounding vegetation types and land uses), macrohabitat (cottonwood forest patch characteristics), and microhabitat (local vegetation characteristics). Management implications from the spatial scale paper (Saab 1999) are included in the last section of this report to provide one reference for managers.

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ABSTRACT

More than any other habitat in western North America, arid-land riparian woodlands are centers of high diversity and abundance of birds. Because these habitats are fragmented and limited in distribution, western riparian birds might be particularly vulnerable to human-caused disturbances. During a four-year study, I examined the influences of land-use practices in relation to cottonwood forest-patch dynamics on bird community and vegetative characteristics in southeastern Idaho. Patterns in bird community characteristics of 34 species, relative abundances of individual species and nest guilds, and vegetation structure were compared among three land uses (areas managed for livestock grazing [grazed], areas managed as campgrounds [recreation], and areas not managed for grazing or recreation but for riparian and wildlife habitat values [unmanaged]), and three patch-size classes (small [<1-3 ha], medium [>3-10 ha], and large [>10-204 ha]). Overall species richness, diversity, evenness, and turnover remained fairly constant among all land uses. On average, species numbers and relative abundance appeared to be most reduced by recreational activities except in large patches managed for recreation. Few differences existed between grazed and unmanaged sites in overall mean number of species or mean number of individuals per survey visit. However, distribution and abundance of individual species, and species grouped by nest layer and nest type, varied significantly among land-use activities and patch sizes. Vegetation structural characteristics within the ground, shrub, and canopy layers were positively correlated with abundance of birds nesting in those layers. Ground-nesting species (Veery and Fox Sparrow) were most susceptible to disturbances created by livestock grazing and were also most sensitive to fragmentation of riparian habitats. Canopy nesters, including cavity-nesting species, responded positively in grazed habitats, while shrub-nesting species tended to decrease with grazing and recreational activities. Significant results of Poisson regression, for 17 of 30 species analyzed, suggested differential effects of land use, patch size, and/or the interaction between the two effects. Relative abundances of 11 species decreased with either grazing or recreation, whereas six species increased with these same activities. Five species (Gray Catbird, Veery, Yellow Warbler, Black-headed Grosbeak, and American Goldfinch) were unaffected by patch size in unmanaged areas but showed significant area effects (increases in probability of occurrence with cottonwood forest area) in grazed and/or recreation sites. Results of my study suggest that conservation of large patches is particularly important where riparian forests are managed for grazing and recreation. Apparently, some species need larger patches of breeding habitat in areas with these disturbances. In addition to evaluating the effects of local land-use practices on habitat relationships of breeding birds, I examined the importance of landscape patterns to habitat use by birds. Among three spatial scales (landscape, macrohabitat, and microhabitat), landscape features were the most important and frequent predictors of distribution and abundance for most bird species and for predicting high species richness of native avifauna. Thus, surrounding landscape features should be a primary consideration for managing riparian habitats and selecting riparian reserve areas.

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EXECUTIVE SUMMARY

More than any other habitat in western North America, arid land riparian forests are centers of high diversity and abundance of birds. Because these habitats are naturally fragmented and limited in distribution, western riparian birds might be particularly vulnerable to human-caused disturbances. During a four-year (1991-1994) research project, we examined patterns of habitat use by breeding birds in cottonwood riparian forests in relation to land-use practices, forest-patch dynamics, and spatial scale. Bird distribution and abundance, and vegetation characteristics were quantified on 57 cottonwood forest patches, ranging in size from 0.40 ha - 205 ha, along 100 km of the South Fork of the Snake River in southeastern Idaho. The goal of this work was to provide information to managers on habitat features that are necessary for the long-term persistence of migratory landbirds breeding in cottonwood forests.

Factors potentially influencing habitat selection in relation to livestock grazing, recreational campgrounds, and forest-patch dynamics were analyzed in discrete categories by land use (grazed, recreation, and unmanaged), and patch size (small [< 1- 3 ha], medium [> 3 - 10 ha], and large [> 10 ha]). On average, species richness and relative abundance were most reduced in recreational campgrounds, except in large patches managed for recreation. Ground-nesting Neotropical migrants were most susceptible to disturbances created by livestock grazing and were also most sensitive to fragmentation of riparian habitats. Five species were unaffected by patch size in unmanaged areas but showed but showed significant area effects (increases in probability of occurrence with increases in forest area) in grazed and/or recreation sites. These results suggest that some species may need larger patches of breeding habitat in areas with these disturbances.

To examine the importance of spatial scale to habitat use, a hierarchical approach was used at three scales: microhabitat (local vegetation characteristics), macrohabitat (features of cottonwood forest patches), and landscape (composition and patterning of surrounding vegetation types and land uses). The surrounding landscape changed from a valley surrounded by mountains on the upstream end of the study area, a narrow canyon adjacent to upland natural vegetation in the middle section, to a wide, flat floodplain dominated by agriculture on the downstream end. The best predictors of high species richness of native birds were natural and heterogeneous landscapes, large cottonwood patches, close proximity to other cottonwood patches, and microhabitats with relatively open canopies. Landscape features were the most important and frequent predictors of distribution and abundance for most bird species, while macrohabitat and microhabitat were of secondary importance. Thus, landscape features should be a primary consideration for management of riparian habitats and selecting riparian reserve areas.

INTRODUCTION

Evidence of widespread population declines among many species of Neotropical migratory birds (Terborgh 1989, Hagan and Johnston 1992, Martin and Finch 1995, Rappole 1995) has intensified interest in avian conservation among scientists and land managers. The concern for Neotropical migrants (landbirds that breed mainly in temperate North America and winter primarily south of the United States-Mexico border [Finch and Stangel 1993]) first became heightened when Robbins et al. (1989a) reported that 75 percent of forest-dwelling migrants in eastern North America experienced population declines during the 1980s. Several human-caused factors (e.g, fragmentation and degradation of habitats) were indicated as operating to adversely affect populations of these species.

Although concern for these species originated from population monitoring of avifauna in the species-rich deciduous forest of the eastern United States, densities of breeding migrants are probably much higher in riparian habitats of western United States (Carothers and Johnson 1975, Ohmart and Anderson 1986, Knopf et al. 1988a, Finch 1991, Krueper 1993). These habitats comprise less than 1 percent of the landscape in the arid western United States, yet more species of breeding birds are found in this limited habitat compared to the more extensive surrounding uplands (e.g., Mosconi and Hutto 1982, Knopf et al. 1988a, Finch 1991, Saab and Groves 1992). Because of the presence of free water, riparian habitats in western North America have been greatly exploited and have suffered a century of degradation from livestock grazing, agriculture, water diversion, recreation, and other land-use activities (Thomas et al. 1979, Blakesley and Reese 1988, Sedgwick and Knopf 1989, Rood and Heinze-Milne 1989, Bock et al. 1993a, Malanson 1993, Ohmart 1994, Knight and Gutzwiller 1995, Saab et al. 1995). Western riparian areas are among the most threatened habitats on the continent because they are favored for these many land uses (cf. Terborgh 1989).

Because western riparian habitats are fragmented and limited in distribution, the total population numbers of migratory birds in these habitats probably are much smaller than those of migrants in woodlands of eastern North America (Terborgh 1989, Finch 1991). Consequently, western migratory landbirds may be particularly vulnerable to degradation of riparian woodlands. Thus, protection of existing healthy riparian woodlands and restoration of degraded or destroyed riparian systems should be a high priority for bird conservation in the western United States. Knowledge of bird responses to various land-use activities can provide critical insights into understanding and sustaining the integrity of riparian ecological systems.

Two major land-use activities altering the quality and quantity of riparian habitats at local, landscape, and regional scales are livestock grazing and recreational activities (e.g., Malanson 1993, Knight and Gutzwiller 1995, Saab et al. 1995). These activities may act alone to influence the persistence of landbirds, but it is also probable that they affect species by interacting together and with other forces (such as habitat fragmentation) in synergistic and cumulative ways.

Habitat fragmentation has been defined as "simply the disruption of continuity," which allows it to apply to any spatial scale (Faaborg et al. 1995). Fragmented habitats result in both quantitative and qualitative losses of habitat for species originally dependent on that habitat (Temple and Wilcox 1986). As a consequence, species abundance and diversity often decline and losses are greatest in smaller fragments (see Askins et al.1990).

Disturbances created by livestock grazing and recreational activities could exacerbate losses of plant and animals occupying small habitat fragments. Birds generally do not respond to the presence of grazing livestock but to the indirect impacts on vegetation as a result of grazing (Bock and Webb 1984). Grazing activities affect riparian habitats by altering, reducing, or removing vegetation, and by actually eliminating riparian areas through channel widening or lowering the water table (cf., Platts 1991, Mulchunas and Lauenroth 1993, Fleischner 1994).

Recreational activities not only affect birds through the indirect impacts of habitat modification but also directly by the presence of humans (Knight and Cole 1995a). Historically, the perception has been that outdoor recreation posed little environmental threat compared to extractive uses of natural resources such as timber harvest and livestock grazing (Flather and Cordell 1995). However, recreationists can degrade the land, water, and wildlife resources that support their activities by simplifying plant communities, increasing animal mortality, displacing and disturbing wildlife, and distributing refuse (Boyle and Samson 1985). Human-induced disturbance can have significant negative effects on breeding success by causing nest abandonment and increased predation (Hockin et al. 1992).

Several studies have evaluated the effects of cattle grazing on breeding bird communities in riparian habitats of western North America (e.g., Mosconi and Hutto 1982, Sedgwick and Knopf 1987, Knopf et al. 1988b, Schulz and Leininger 1991), whereas I am aware of only two studies that have examined the impacts of recreational activities on riparian avifauna (Aitchison 1977, Blakesley and Reese 1988). No prior study has investigated the influences of grazing and recreation in tandem, or in relation to forest patch dynamics, and these conditions often occur together in the western United States.

In this study, I evaluated bird and vegetation characteristics in cottonwood riparian forests, along the Snake River in Idaho, under three types of land use: (1) areas managed for cattle grazing [grazed]; (2) areas managed as campgrounds [recreation]; and (3) areas not managed for grazing or recreation but for riparian and wildlife habitat values [unmanaged]. Although landscape features surrounding forest patches are very important in explaining patterns of habitat use by breeding birds (e.g., Saab 1999, Freemark et al. 1995) and should be considered for certain types of analyses (particularly when making regional comparisons), my goal here was to examine, within a microhabitat (local) scale, the relative influence of grazing, recreation, and patch size on the distribution and abundance of breeding birds in riparian forest patches. Specific questions were: (1) How do bird species composition, richness, and abundance differ among lands managed for livestock grazing, recreation, and areas with little direct management? (2) Does riparian-forest patch size influence bird responses to these land-use practices? (3) Do bird species diversity, evenness, and turnover differ among land uses ? (4) Which individual bird species and nest guilds are favorably or negatively affected by the different land uses and patch sizes? and (5) Are structural patterns in vegetation layers among land uses and patch sizes similar to abundance patterns of birds occupying those vegetation layers?

STUDY AREA AND METHODS

Study Area Description

The study area encompassed the cottonwood riparian forests along 100 km of the South Fork of the Snake River (South Fork) in southeastern Idaho (Fig.1). Elevation ranges from 1700 m on the upstream end to 1460 m on the downstream end (confluence with the Henry's Fork of the Snake River). The climate is characterized by relatively low annual precipitation (550 mm), most of which comes in the form of snowfall during winter months.

The surrounding landscape is dominated by upland natural vegetation on the upstream end, and by agriculture on the downstream end (Saab 1992). Some cottonwood forests are naturally fragmented in the upper portions of the river, while others are fragmented as a result of agricultural development, especially downstream. Cottonwood fragments ranged in size from < 1 ha to > 200 ha.

The streamside vegetation is dominated by narrowleaf cottonwoods (*Populus angustifolia*) in the canopy, with the woody subcanopy/understory vegetation composed primarily of red-stemmed dogwood (*Cornus stolonifera*), and lesser amounts of thin-leaved alder (*Alnus incana*), water birch (*Betula occidentalis*), willows (*Salix* spp.), Rocky Mountain juniper (*Juniperus scopulorum*), silverberry (*Elaeagnus commutata*), chokecherry (*Prunus virginiana*), and hawthorne (*Crataegus* spp). The stream corridor is managed for irrigation, recreation, and livestock grazing.

Study Site Selection

Fifty-seven cottonwood forest patches (i.e., study sites) were located along a 100-km section of the South Fork. More than half (54%) of the cottonwood patches were created as a result of agricultural development; the remaining patches were created naturally by river channels. Selection criteria included land use activities, age and size classes of cottonwood patches, and isolation of cottonwood stands. Cottonwood forest patches were determined by delineating breaks in the forest canopy that were at least 100 m in width. All sampling areas were located in mature cottonwood stands. Each study site was managed for one of three types of land uses: (1) cattle grazing, (2) recreation campgrounds, and (3) unmanaged [areas not managed for grazing or recreation with little human use]. Land-use classes were selected on the basis of management activities. Actively managed sites were located in cottonwood patches where at least 75% of the area was used by livestock or public recreation. Unmanaged areas were not managed for livestock grazing or recreation and met the following criteria: (1) vegetation relatively undisturbed, (2) no obvious recent disturbance by humans, and (3) free from livestock grazing for at least three years. Cottonwood forests exclusive of livestock grazing and outside of designated campgrounds were managed for their riparian and wildlife habitat values.

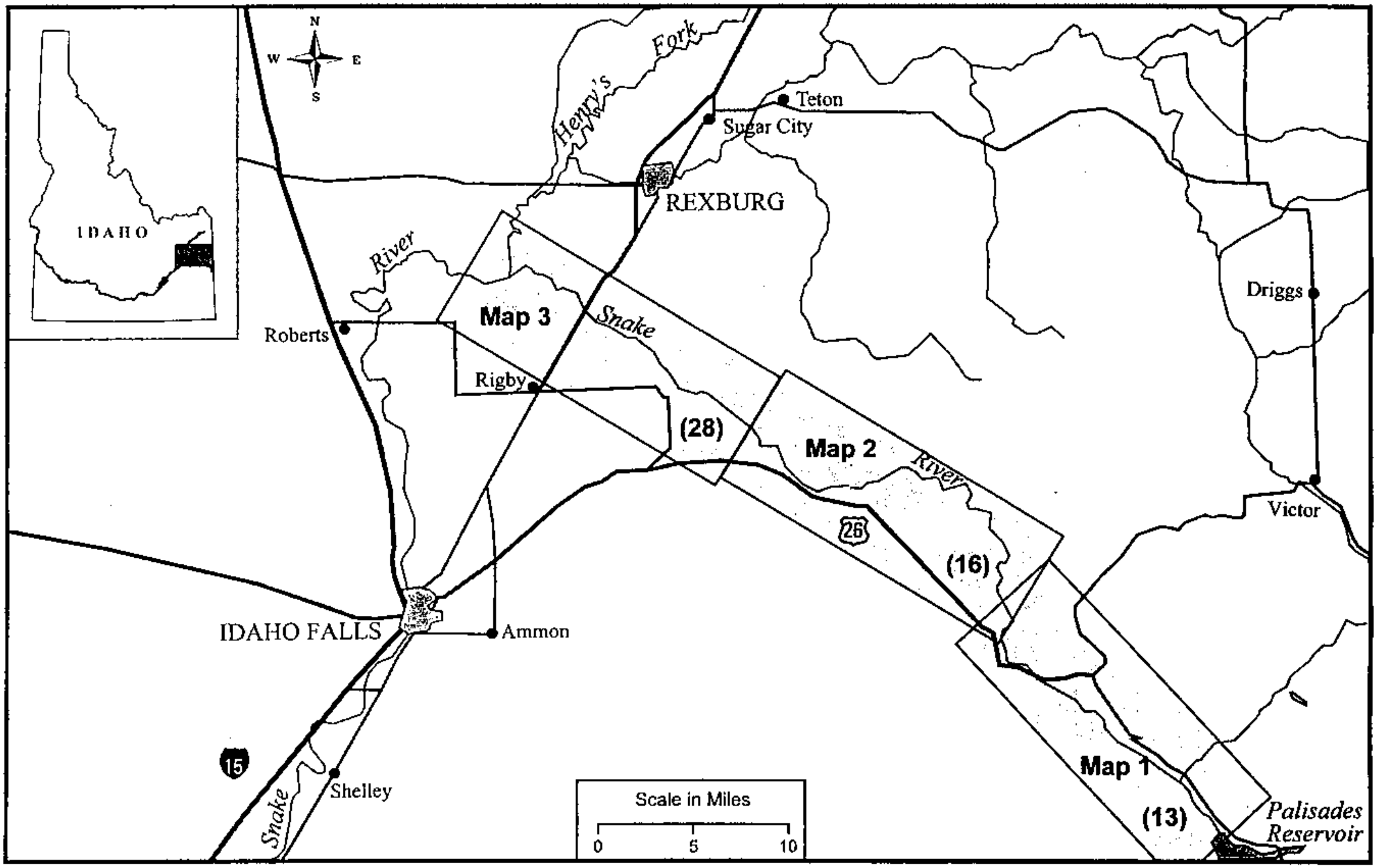

Figure 1. Location of the cottonwood riparian study area. Numbers in parentheses are the number of cottonwood patches in each map section that were sampled from 1991 - 1994.

Once I evaluated all cottonwood patches that met these criteria, sites were selected at random, except for recreation sites in small patches and all land uses in the large size class, where all available sites were selected for study. I avoided selecting sites managed for both grazing and recreation except for one large-patch recreation site that was grazed in autumn after the growing season and after migratory birds departed the study area. Riparian habitats grazed during autumn months (as compared to other seasons of the year) apparently have the least impact on breeding birds (Saab et al. 1995).

Table 1. Number of cottonwood patches and sampling stations within cottonwood patches used for surveying birds with point counts and measuring vegetation.

		Land Use		
Patch Size (ha)	Grazed	Recreation	Unmanaged	Totals
		No. Patches/*Stations*		
Small (<1-3)	13 / *17*	2 / *3*	13 / *21*	28 / *41*
Medium (>3-10)	3 / *8*	3 / *10*	9 / *25*	15 / *43*
Large (>10-200)	5 / *26*	4 / *20*	5 / *23*	14 / *69*
Totals	21 / *51*	9 / *33*	27 / *69*	57 / *153*

Each study site fit one of three patch-size classes (small, medium, and large; <1-3 ha, >3-10 ha, and >10-204 ha, respectively), resulting in nine sampling categories (Table 1). Size classes of cottonwood patches were based on the effects of patch size on breeding birds in riparian forests (Stauffer and Best 1980, Gutzwiller and Anderson 1987) and in forests fragmented by agriculture (Loman and von Schantz 1991). Within each size or land-use class (e.g., small grazed; see Table 1) there were at least 33 permanently marked stations, 150 m apart, from which birds were surveyed and vegetation was measured. This resulted in 153 sampling stations, with 84 stations in managed areas and 69 in unmanaged habitats.

Sites managed for livestock were variously grazed by cattle, and the timing and intensity of grazing were not controlled. For the previous five years and during the study, most (57%) sites were grazed throughout the growing season (May-September); some (29%) were grazed only in spring, summer, or fall; and a few (14%) were grazed on a rest-rotation basis (cf. Saab et al. 1995). Grazing intensity ("light," "moderate," or "heavy," based on the percentage of herbaceous vegetation consumed by livestock [Society of Range Management 1989]), varied among sites but was usually moderate (57%), often heavy (38%), and occasionally light (5%).

Campsites had been established for at least 10 years on all areas managed for recreation, and these sites were characterized by patches of trampled vegetation and/or bare soil. Most campgrounds were considered primitive because few or no facilities existed for recreationists. Recreational activities were primarily camping, day hiking, and fishing but at the most heavily used campgrounds (two of nine sites), recreation also included off-road driving. Most campgrounds were used only on weekends from late-spring throughout the summer. The number of recreationists varied among sites. On the two largest (> 10 ha) and most heavily used campgrounds, an average of 37.65 ($\pm$ 8.83 [1 SE]) recreationists were camped on a typical summer weekend (U.S.D.I. Bureau of Land Management, unpublished data).

Bird Surveys

Relative bird abundance was quantified using point-count surveys (Ralph et al. 1993) in 40-meter fixed-radius circular sampling stations (cf. Szaro and Jakle 1985) that were placed at least 20 m from a forest edge or in the center of the 57 cottonwood patches. The number of circular sampling stations varied depending on the area of the cottonwood patch, from one on sites of less than 4 ha to as many as 6 on a site more than 200 ha. On sites with more than one sampling station, station centers were separated by at least 150 meters. A total of 153 stations were sampled in this study. Observers surveyed birds for 10 minutes per visit at each station. Each station was visited twice during the 1991 breeding season, and three times during 1992-1994, from 15 May to 15 July. Bird counts were conducted by two people during 1991, and by three people during other years (1992-1994). Two of the same observers conducted point counts in three of four years. Each observer visited every station in each season in an attempt to minimize observer effects (Verner 1985).

Bird surveys were conducted between 06:00 and 11:00 and were confined to days with good weather (wind less than 20 mph and light or no precipitation). Stations were sampled at different times in the morning during the breeding season to reduce time-of-morning effects (Ralph et al. 1993). To reduce the bias of surveying at different periods of the breeding season, the first survey was conducted 15 May-4 June, the second survey from June 5-24, and the third survey from 25 June-15 July of each year. The total number of species, the number of individuals of each species, and the total number of individuals were recorded for each circular sampling station. Species and individuals recorded as flying over survey stations and species not breeding in the study area were excluded from analyses.

Thirty-four bird species known to nest in the study area were used in making estimates of total species richness and overall abundances for small landbirds (Appendix 1, Appendix 2). Thirty of those species were used for all analyses and were recorded in a minimum of 12 cottonwood patches and at least 14 sampling stations.

Habitat Measurements

Vegetation measurements were collected during 1991 and 1992 at the 153, 40-meter radius (0.5 ha) circular sampling stations used for bird surveys (see Table 1). Within each 0.5 ha sampling station, estimates of vegetation structure and composition were made in four 5-m-radius subcircles (0.008 ha). The initial subcircle was located in the center of the sampling station. The center of the next subcircle was located at a random compass direction and a fixed distance of 29 m from the station center. Each of the two remaining subcircles were positioned 120° from the first subcircle (cf. Ralph et al. 1993).

Stem densities of trees and shrubs were recorded by species and diameter size class at breast height. Woody vegetation was grouped into diameter size classes as follows: $\leq$2 cm, >2-5 cm, >5-8 cm within the 5-m-radius subcircle; and >8-23 cm, >23-38 cm, and >38 cm within a 11.3-m-radius circle extended from the 5-m-radius plot. Tree canopy was measured by using a densiometer at the center of each subcircle.

Ground cover was estimated on the 5-m radius subcircles by using an ocular tube (James and Shugart 1970). Ten readings were taken along transects using tape measures oriented parallel to the stream channel and the other perpendicular, such that they crossed at the center of the subcircle. Ground cover was estimated as percentages of either shrub, herbaceous, bare ground or litter based on frequency of occurrence at 2-m intervals along the tapes.

Data Analysis

Bird species richness, abundance, species diversity and eveness, and local species turnover were compared among grazed, recreation, and unmanaged cottonwood sites. Species diversity was calculated using the Shannon index of diversity [$H' = -\sum p_i \ln p_i$], where p_i is the proportion of individuals found in the ith species, and evenness [$E=H'/\ln S$], where S is total number of species (Magurran 1988). Local species turnover was determined by averaging the year to year change in species composition recorded over all sampling stations within a single cottonwood patch for three time periods: from 1991 to 1992, from 1992 to 1993, and from 1993 to 1994 (cf. Haila et al. 1993). Bird survey and vegetation data were combined by cottonwood patch to provide one sample per patch.

A two-way and three-way analysis of variance (ANOVA) or MANOVA for an unbalanced design (PROC GLM, SAS Institute, Inc. 1990) were the primary tests used for statistical comparisons of relative bird abundance and richness, bird species turnover rates, abundance of bird species grouped by nest layer (placement of nests as either ground, shrub, or canopy) and nest type (cavity or open), canopy coverage, woody-plant stem densities, and ground cover among land uses, patch sizes and in some cases among years. To test differences in species richness and abundance between various combinations of land use and patch size, I created an independent variable that had a level for each combination of the original independent variables (three land-use types and three patch-size classes) by concatenation (Cody and Smith 1991) and then performed a one-way ANOVA and multiple range tests for the main effects. Continuous variables used in (M)ANOVAs were tested for univariate normality, and arcsine (percentage data) or log (count data) transformed if necessary. Following transformation, all variables were normally distributed (Shapiro-Wilkes statistic, $P > 0.10$, SAS PROC UNIVARIATE). Type III Tests (SS [sums of squares] or SS&CP[cross products] matrices) were used for all (M)ANOVAs.

Wilk's Lambda was selected as the MANOVA test statistic. All multiple comparisons of main effects were computed with Tukey studentized range tests (SAS Institute, Inc. 1990). To determine if abundance of species grouped by nest layer was related to vegetation structure, vegetation layer measurements (ground cover, shrub densities, and canopy cover) were correlated with species' abundances in each nest layer (ground, shrub, canopy) using Spearman ranks.

Separate analyses on relative abundance of 30 individual species were completed for each species with numbers of detections large enough to fit regression models that included effects of land use and patch size (Table 2). Year was not included in the individual species models because few species had large enough sample sizes for valid results. Poisson regression (SAS/INSIGHT, SAS Institute, Inc. 1993) was used to test statistical differences in the mean number of detections (response variable) for each species per point count visit among land uses and patch sizes (the two main effects) and the interaction between the main effects. The interaction effect was not tested for seven species with small numbers of observations in each combination of land use/patch size, and a reduced model was fitted using only the main effects (Table 2). A Type III Wald Chi-square was the test statistic for the Poisson Regression. Poisson regression was appropriate for analyses of individual species because the response variable represented counts, including many zeros. For species in which the analysis showed a significant interaction effect between land use and patch size or a significant patch size effect, logistic regression was used to aid in examining the relationship between area (ha) and probability of the species' occurrence (Robbins et al. 1989b) within each type of land use. In the logistic regression analysis, an index of occurrence was calculated for each species for each cottonwood patch to serve as the dependent variable. In this calculation, the dependent variable assumes a value of 0 if the species was not detected on a point count visit to the sampling station and 1 every time it was detected on a single visit. Most sampling stations were visited 11 times over four breeding seasons, and the average number of occurrences per sampling station within a cottonwood patch was used to calculate a species' predicted probability of incidence within a cottonwood patch. Alpha-levels were set at 0.05 for all statistical tests.

Table 2. Mean number of detections per point count visit ($\pm$ 1 standard error) for each species preceded by the number of sites [ ] in which each species was recorded. The interaction effect between land use and patch size was not tested (NT) for species with few observations (see text for explanation). Numbers under each type of land use or patch size are the number of cottonwood patches sampled for that group. NS = not significant. For species with significant effects, different letters indicate that corresponding means are significantly different ($p \leq 0.05$) within land uses or within patch sizes. Common names of species' acronyms are listed in the Appendix 1.

Species	Land Use				Patch Size				Interaction
	Grazed (21)	Recreation (9)	Unmanaged (27)	*P* -value	Large (14)	Medium (15)	Small (28)	*P*-value	*P* -value
AMKE	[7] 0.08(0.03)	[4] 0.05(0.02)	[13] 0.06(0.02)	NS	[8] 0.07(0.02)	[7] 0.04(0.02)	[9] 0.07(0.02)	NS	NT
MODO	[20] 0.61(0.09)a	[8] 0.19(0.03)b	[22] 0.25(0.03)b	<0.001	[14] 0.37(0.04)a	[11] 0.22(0.04)b	[25] 0.44(0.07)a	0.034	0.002
RNSA	[14] 0.11(0.02)	[8] 0.16(0.03)	[21] 0.18(0.03)	NS	[14] 0.21(0.03)	[12] 0.15(0.03)	[17] 0.12(0.03)	NS	NS
DOWO	[15] 0.11(0.02)	[5] 0.04(0.01)	[18] 0.07(0.01)	NS	[15] 0.06(0.01)	[9] 0.09(0.02)	[14]0.08(0.02)	NS	NT
NOFL	[17] 0.23(0.03)	[8] 0.24(0.04)	[25] 0.18(0.03)	NS	[15] 0.26(0.04)	[12] 0.18(0.02)	[23] 0.19(0.03)	NS	NS
WWPE	[17] 0.23(0.03)	[4] 0.08(0.03)	[19] 0.20(0.03)	NS	[12] 0.20(0.03)	[9] 0.15(0.03)	[19] 0.22(0.03)	NS	NS
DUFL	[9] 0.06(0.02)a	[6] 0.04(0.01)b	[11] 0.03(0.01)b	0.002	[11] 0.05(0.01)a	[8] 0.08(0.02)a	[7] 0.02(0.01)b	0.004	NT
BBMA	[20] 0.53(0.09)a	[6] 0.30(0.09)ab	[20] 0.23(0.04)b	0.015	[15] 0.21(0.03)	[10] 0.32(0.09)	[21] 0.44(0.07)	NS	0.012
AMCR	[10] 0.11(0.03)	[5] 0.19(0.06)	[12] 0.10(0.02)	NS	[10] 0.17(0.05)	[8] 0.10(0.02)	[27] 0.09(0.02)	NS	NS
BCCH	[18] 0.20(0.03)a	[8] 0.24(0.05)ab	[25] 0.35(0.04)b	0.047	[15] 0.33(0.03)	[13]0.35(0.06)	[23] 0.22(0.04)	NS	NS
HOWR	[19] 1.15(0.09)a	[6] 0.17(0.04)b	[14] 0.46(0.06)b	<0.001	[13] 0.56(0.07)	[9] 0.46(0.08)	[17] 0.82(0.09)	NS	NS
GRCA	[9] 0.07(0.02)	[7] 0.15(0.04)	[21] 0.15(0.02)	NS	[12] 0.18(0.02)a	[11] 0.08(0.01)b	[14] 0.11(0.03)ab	0.025	NS
AMRO	[21] 1.34(0.09)	[8] 0.95(0.10)	[25] 1.19(0.08)	NS	[15] 1.08(0.09)	[13]1.11(0.10)	[26] 1.33(0.08)	NS	NS
VEER	[7] 0.11(0.03)a	[7] 0.25(0.05)a	[20] 0.48(0.05)b	0.048	[14] 0.38(0.05)	[8] 0.33(0.06)	[12] 0.28(0.05)	NS	<0.001
CEWA	[13] 0.20(0.04)	[7] 0.29(0.04)	[23] 0.31(0.05)	NS	[15] 0.24(0.03)	[13] 0.29(0.03)	[15] 0.27(0.05)	NS	NS
EUST	[19] 1.25(0.20)a	[4] 0.13(0.06)b	[21] 0.35(0.07)b	0.039	[13] 0.25(0.06)	[9] 0.18(0.05)	[22] 1.07(0.16)	NS	0.038

Table 2. Continued

Species	Land Use: Grazed (21)	Land Use: Recreation (9)	Land Use: Unmanaged (27)	*P*-value	Patch Size: Large (14)	Patch Size: Medium (15)	Patch Size: Small (28)	*P*-value	Interaction *P*-value
WAVI	[12] 0.35(0.05)a	[8] 1.00(0.09)b	[22] 0.49(0.05)a	<0.001	[15] 0.78(0.06)a	[12] 0.72(0.08)a	[15] 0.27(0.04)	<0.001	NS
YEWA	[20] 2.16(0.11)	[8] 2.13(0.17)	[27] 2.71(0.10)	0.049	[15] 2.67(0.12)a	[13] 2.37(0.14)ab	[27] 2.34(0.10)b	0.045	NS
YRWA	[11] 0.08(0.02)	[8] 0.20(0.04)	[13] 0.06(0.01)	NS	[14] 0.15(0.03)	[11] 0.13(0.02)	[7] 0.04(0.01)	NS	NS
MGWA	[6] 0.02(0.01)	[8] 0.15(0.04)	[12] 0.06(0.02)	NS	[10] 0.03(0.01)	[8] 0.10(0.03)	[8] 0.05(0.02)	NS	NS
YBCH	[3] 0.02(0.01)a	[2] 0.01(0.01)a	[7] 0.05(0.02)b	0.035	[6] 0.07(0.03)	[4] 0.04(0.02)	[2] 0.02(0.01)	NS	NT
WETA	[10] 0.05(0.01)	[7] 0.05(0.01)	[10] 0.03(0.01)	NS	[10] 0.06(0.02)	[8] 0.05(0.02)	[9] 0.03(0.01)	NS	NS
BHGR	[14] 0.13(0.02)	[8] 0.24(0.04)	[24] 0.23(0.03)	0.032	[14] 0.25(0.02)	[12] 0.20(0.03)	[20] 0.17(0.03)	0.049	NS
LZBU	[10] 0.06(0.02)a	[4] 0.07(0.02)ab	[20] 0.12(0.02)a	0.038	[11] 0.09(0.02)	[9] 0.09(0.02)	[14] 0.09(0.02)	NS	NT
NOOR	[20] 0.53(0.06)	[8] 0.25(0.05)	[26] 0.41(0.04)	NS	[15] 0.33(0.04)	[12] 0.35(0.05)	[27] 0.52(0.05)	NS	<0.001
BHCO	[19] 0.46(0.06)	[7] 0.17(0.04)	[24] 0.45(0.04)	NS	[15] 0.39(0.04)	[13] 0.36(0.05)	[22] 0.45(0.05)	NS	NS
CAFI	[9] 0.03(0.01)	[4] 0.03(0.02)	[6] 0.02(0.01)	NS	[10] 0.04(0.01)	[5] 0.02(0.01)	[4] 0.02(0.01)	NS	NT
AMGO	[21] 0.92(0.12)a	[8] 0.47(0.08)b	[27] 0.92(0.07)a	0.008	[15] 1.05(0.08)a	[13] 0.70(0.09)b	[28] 0.82(0.09)b	<0.001	0.003
FOSP	[3] 0.01(0.004)a	[4] 0.08(0.03)ab	[15] 0.15(0.03)b	<0.001	[10] 0.09(0.02)	[5] 0.07(0.02)	[7] 0.10(0.03)	NS	NT
SOSP	[13] 0.32(0.05)a	[9] 0.75(0.09)b	[24] 0.78(0.07)b	<0.001	[13] 0.45(0.05)a	[12] 0.63(0.07)ab	[21] 0.70(0.08)b	0.002	NS